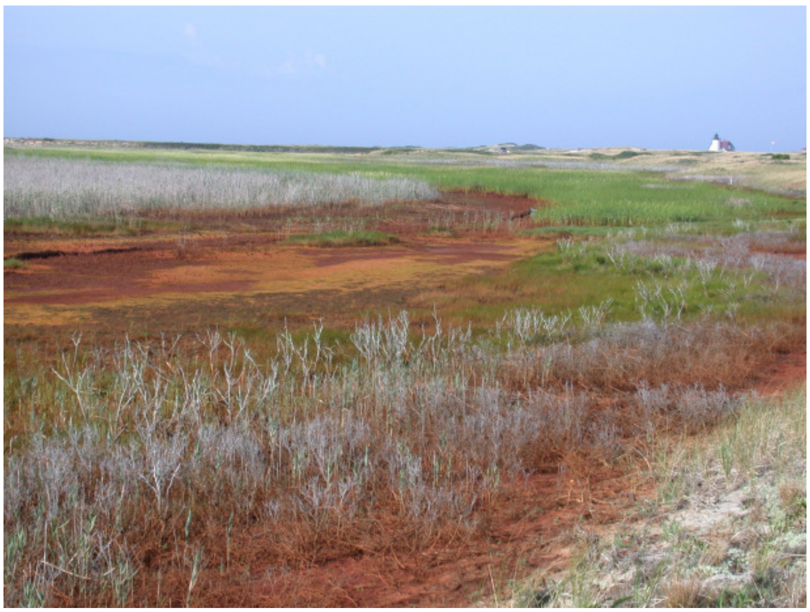
2004 VEGETATION MONITORING REPORT FOR SALT MARSH RESTORATION PROJECTS



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EXECUTIVE SUMMARY

In 2003, restoration of East Harbor, formerly a salt marsh lagoon but impounded for > 130 years, was attempted by establishing an open connection to Cape Cod Bay through an existing culvert. Since the opening, salinities in East Harbor proper have increased nearly five-fold although in the main basin tide-driven fluctuations in water level remain negligible. Emergent macrophyte vegetation, largely consisting of *Phragmites australis* (common reed) and *Typha angustifolia* (narrow-leaved cattail), has exhibited dieback, but this response has been limited to an area immediately upstream from the point of seawater entry. Moreover, native salt marsh taxa have not become established. As such, restoration of peripheral marsh communities may demand an increased capacity for tidal exchange coupled with active plantings/seeding of target species. In contrast, submersed aquatic vegetation (SAV) such as *Zostera marina* (eelgrass) and, to a much larger extent, *Ruppia maritima* (widgeongrass), have proliferated rapidly. Uptake of nutrients and sediment stabilization by these seagrasses should result in improved water clarity, as has already been observed in the northwest cove of East Harbor, which itself will enhance conditions for further seagrass growth.

The vegetation of Hatches Harbor continues to exhibit significant responses to incremental restoration of tidal flow, which was begun approximately 6 years ago. The latest increase in seawater exchange (October 2003) resulted in the creation of even more wetland area as evidenced by the recent mortality of salt-intolerant wetland and upland plants, coinciding with new appearances of native salt marsh species. The *Phragmites* population in Hatches Harbor continues to shift away from the main tidal creeks toward the edge of the basin. This has allowed salt marsh vegetation to expand their range and a substantial portion of the formerly-restricted marsh now resembles a “normal” salt marsh landscape.

CONTENTS

Vegetation monitoring conducted in the salt marsh restoration areas (East Harbor and Hatches Harbor) and in unrestricted salt marshes (West End, Nauset, Middle Meadow, Pleasant Bay) in 2002 and 2003 is detailed in the 2003 Salt Marsh Vegetation Annual Report (Smith 2004). The restoration areas were monitored again in 2004 in keeping with the proposed project schedules. This report discusses the new data from the standpoint of total temporal change and includes a new facet of monitoring – submerged aquatic vegetation – in East Harbor.

I. EAST HARBOR: EMERGENT VEGETATION

Introduction

East Harbor (also known as Pilgrim Lake) was originally a back-barrier salt marsh/coastal lagoon with a 1000-ft wide inlet open to Cape Cod Bay. In 1868, the inlet was completely filled, which effectively impounded the system and converted it to a freshwater lake. In 1894, a culvert was installed to manage lake levels. However, a tide gate within the culvert, while allowing water to drain out, prevented any seawater inflow.

As a result of impoundment, ecological and water quality problems have plagued East Harbor for many decades. These include eutrophication, infestations of exotic, invasive flora and fauna, fish kills, and chironomid midge outbreaks. One of the most visually obvious consequences of long-term hydrologic isolation is the existing vegetation. The native species typically found in unimpacted salt marshes are altogether absent as they have been totally replaced by a freshwater community, comprised mostly of narrow-leaved cattail (*Typha angustifolia*) and an exotic strain of the common reed (*Phragmites australis*).

In the interest of restoring East Harbor to its original condition, Cape Cod National Seashore and the Town of Truro permanently opened the tide gate in 2002 to allow for seawater exchange through the existing culvert (a complete chronology of events leading up to the restoration of East Harbor can be found in Appendix I). Monitoring vegetation responses to the reintroduction of seawater is a key component in evaluating the responses of the system.

Methods

A detailed overview of the East Harbor vegetation monitoring program, including objectives, field methodologies, and preliminary results can be found in the 2003 Salt Marsh Vegetation Monitoring Report (Smith 2004).

In May 2002, before any vegetation sampling was conducted, 37 permanent sampling plots ($1m^2$) divided among 4 transects were established in two large areas of peripheral marsh (Figure 1). At the east end of the system, two transects were laid out in a

Phragmites-dominated marsh known as Moon Pond. At the west end, another two transects were placed across a wide peninsula dominated by *Typha*.

The cover of all species within each plot was assessed by visual estimation using cover class ranks of the Braun-Blanquet scale (Braun-Blanquet 1932). The point-intercept method normally used for salt marsh monitoring (Roman et al. 2001) was not used because of difficulties in setting up the plot in extremely dense *Phragmites* and/or poison ivy (*Toxicodendron radicans*). *Phragmites* stem heights and densities were recorded at the end of the 2003 and 2004 growing seasons (September). Because this was not done in 2002, however, the heights of standing dead plants (i.e., over-wintering) that still retained an inflorescence in early 2003 were measured. These represent the previous year's (i.e., 2002) population as older stems rarely bear an inflorescence through two winters. For this report, vegetation data from 2002 (onset of restoration) and 2004 (2 yrs. into restoration) are compared.

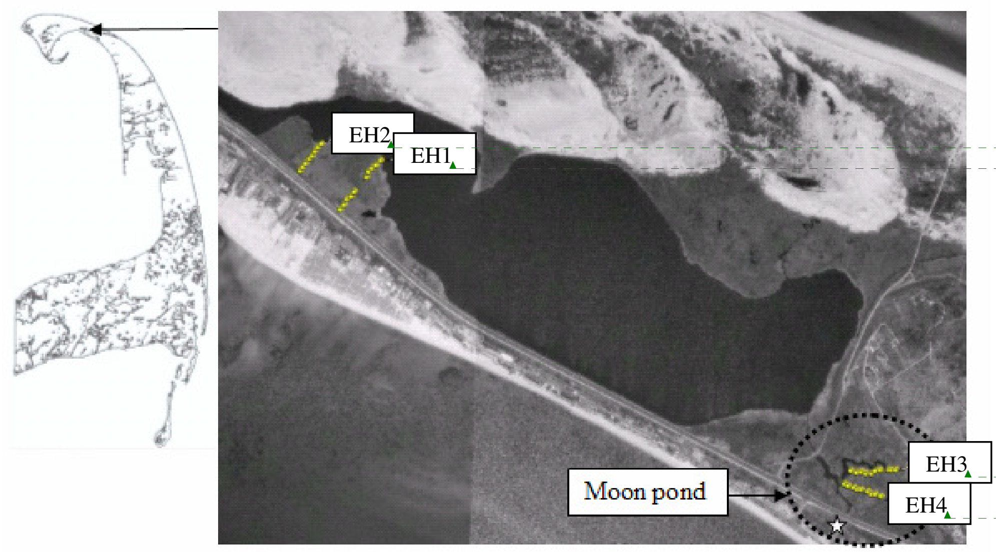

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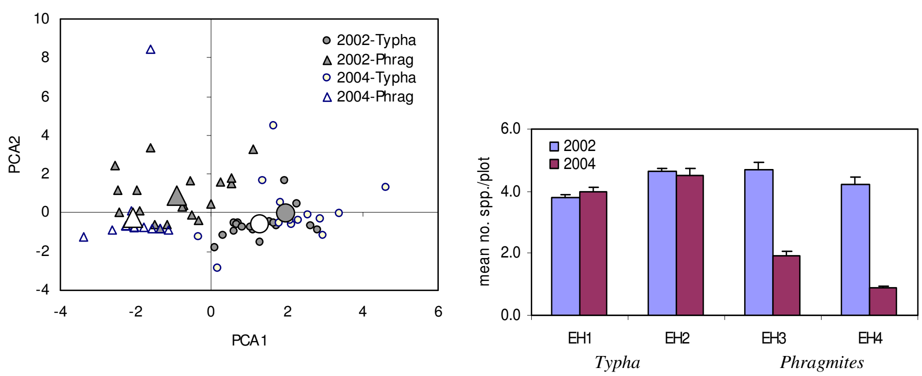
Figure 1. Overview of East Harbor with plot locations along transects 1 through 4 (the *Phragmites*-dominated Moon Pond area is circled; the star indicates the point of seawater entry through an underground culvert open to Cape Cod Bay).

Data analysis - Non-parametric Mann-Whitney U-tests were used to evaluate shifts in cover class rankings of individual species (XLStat ver. 7.5). Principle components Analysis (PCA) of log-transformed percentage values representing the mid-point of cover classes served to illustrate temporal shifts in community composition, while Analysis of Similarities (ANOSIM) tested the statistical significance of community-level responses (Primer ver. 5). For univariate data (e.g., stem height and density), values were log-transformed to improve normality and homoscedasticity and then subjected to Student T-tests to analyze for both marsh-wide and individual plot changes between years.

Results

ANOSIM of vegetation communities indicates that significant shifts in taxonomic composition have occurred during the past two years of restoration (p=0.006) (Figure 2a). The vast majority of this change has occurred along transects 3 and 4 (which are much closer to the point of seawater entry) and is largely the result of declining species diversity. Figure 2b illustrates how the average number of species per plot has significantly decreased since tidal exchange was facilitated. However, it is mostly secondary species that have disappeared while the two dominate taxa – *Typha angustifolia* and *Phragmites australis* – have actually shown significant increases in cover (Table 1). *Chenopodium album* (lambs quarters) has also become more abundant. Although a salt tolerant forb, this species is rare in unrestricted salt marshes within the Seashore.



A. B.

Figure 2. a) PCA of species composition showing movement of individual plots in ordination space over time (*Typha* indicates transects 1&2 while *Phrag* indicates transects 3&4; larger symbols represent centroids of clusters) and b) mean number of vascular plant species by transect in 2002 vs. 2004.

Table 1. List of species showing significant change in cover class values from 2002 to 2004 (asterisk indicate taxa with positive changes).

Comment [MSOffice1]: labeling the transects in fig 1 as T1, T2, etc and then here (in B) labeling them as EH1, EH2, with "Typha" and "Phrag" as sub-labels is kind of confusing. You explain it in the text, but at first look, B seems like its trying show some value for Typha plants in EH1 and EH2, and the same value for Phrag plants in EH3 and EH4. Might be better to refer to the 2 areas with different names (Moon Pond and ?), use consistent transect labels (either T1 or EH1), and point out in the text that the Moon Pond transects are dominated by Phrag and the transects at the other area are dom by Typha.

Species	Jun-02	Aug-04	Change
Calystegia sepium	0.27	0.06	-0.21
Chenopodium album	0.00	0.11	0.11
Eupatorium dubium	0.11	0.00	-0.11
Impatiens capensis	0.38	0.08	-0.30
Rosa virginiana	0.14	0.00	-0.14
Thelypteris palustris	1.59	1.33	-0.26
Toxicodendron radicans	1.62	0.89	-0.73

In the case of *Phragmites*, cover changes were spatially heterogeneous in that increases only occurred along segments of the transect most distant from the point of seawater entry (EH3) (Table 2). These locations have salinities that are high enough to suppress the growth of competing species (> 5 ppt), but low enough to produce little physiological stress on *Phragmites*, which can grow in salinities up to 22 ppt or more (Lissner and Schierup 1997). Conversely, most decreases in cover occurred along transect EH4, where conditions of higher salinities and sulfide concentrations prevail (Table 2).

Table 2. Comparison of *Phragmites* cover class values in August 2003 and 2004 by plot and distance away from culvert (numeric values in plot ID indicate distance away from tidal creek).

Plot	Distance from culvert (m)	cover class Aug 2003	cover class Aug 2004	Change	Aug03 Salinity (ppt)	Aug04 pH	Aug04 sulfide (uM)	Elevation (NGVD)
EH3-060	169	5	5	0	24	6.47	200	0.68
EH3-080	179	3	4	1				0.51
EH3-100	174	5	4	-1	25	6.51	1088	0.58
EH3-120	186	5	5	0				0.58
EH3-140	189	5	5	0	24	6.65	205	0.61
EH3-160	207	5	5	0				0.68
EH3-180	221	4	5	1	23	6.41	1078	0.63
EH3-200	234	5	5	0				0.58
EH3-220	248	5	5	0	21	6.54	493	0.73
EH3-240	263	2	4	2				0.62
EH4-000	113	5	5	0	25	6.29	190	0.69
EH4-020	112	5	5	0				0.74
EH4-040	109	1	1	0	33	6.86	2645	0.55
EH4-060	111	3	2	-1				0.60
EH4-080	114	1	0	-1	32	6.73	3772	0.50
EH4-100	123	5	4	-1				0.62
EH4-120	130	4	4	0	30	6.28	927	0.59
EH4-140	147	5	3	-2				0.56
EH4-160	160	5	5	0	29	6.39	1254	0.65

Despite some positive changes in *Phragmites* growth in the horizontal plane, reductions in stem heights suggest a decrease in vertical growth in most plots (Figure 3). Because stem height contributes much more to plant dry weight per unit area than does stem density (Thursby et al. 2002), this should translate to significant reductions in *Phragmites* biomass.

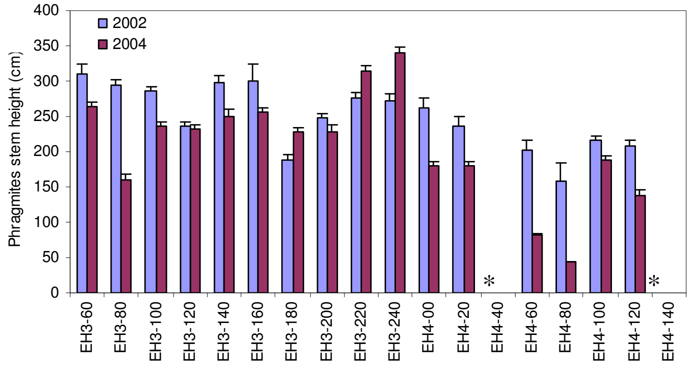


Figure 3. Comparison of *Phragmites* mean stem heights in 2002 vs. 2004 within transect 3 and 4 plots (numbers after the dash in plot names indicate distance from the main tidal creek; 2002 values are based on June 2003 measurements of dead standing stems bearing an inflorescence; asterisks indicate where data are not available).

While values for average stem heights could be obtained by measuring the remaining over-wintering stems bearing an inflorescence in 2003 (i.e., previous year's growth), reliable values for stem densities could not. This is due to the fact that an unknown number of stems break during winter storms. Thus, any counts of stems with intact flower heads would be an underestimate of the true population size. Compared to 2003, when live stems were counted at the end of the growing season, stem densities in 2004 showed variable change, with increases along most of transect 3 and decreases over most of transect 4.

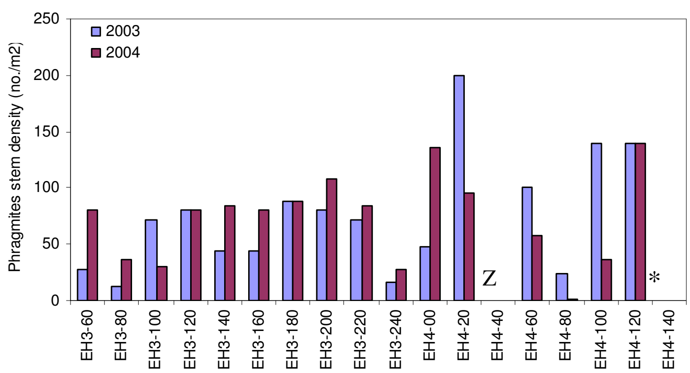


Figure 5. Mean stem densities of *Phragmites* in transects 3 and 4 (Z indicates where Phragmites stem density was 0 for both years; asterisk indicate where data are not available).

In a qualitative sense, oblique angle aerial photography acquired in August 2004 provides a good snapshot of vegetation changes that have occurred in and around Moon Pond. Zones of dead *Typha* and various freshwater shrubs appear as dark gray-black areas, while stunted *Phragmites* shows up as light brown. Most of the green areas are vigorously-growing *Phragmites* populations.

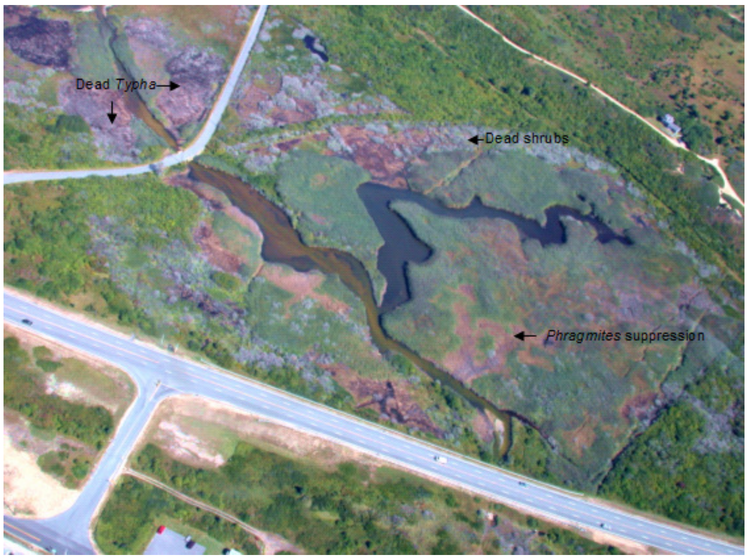


Figure 6. Photograph of Moon Pond area of East Harbor (August 2004) showing plant mortality and *Phragmites* growth suppression.

Discussion

Vegetation in the emergent marshes of East Harbor has undergone substantial change since the culvert connecting Moon Pond with Cape Cod Bay was opened to tidal flow in 2002. The largest responses have occurred in the Moon Pond area, where seawater first enters the system. Here, virtually all salt-intolerant taxa have disappeared. In some locations *Phragmites* is showing signs of stress, as evidenced by a reduction in height and stem densities (or complete disappearance as was the case for one plot). In other locations, however, *Phragmites* continues to thrive and may actually be benefiting from a reduction in interspecific competition (Lenssen et al. 2003). This is similar to what has been observed in another tidal restoration project within the Seashore (Hatches Harbor) where *Phragmites* is encroaching upon freshwater wetland communities bordering the upland transition zone (Smith et al. 2003).

Although the reintroduction of seawater has resulted in an increase from < 5 to 25 ppt within East Harbor proper, the current level of tidal amplitude is probably insufficient to influence the emergent vegetation of the bordering marshes. With the exception of Moon Pond, where responses are much more pronounced, the decline of freshwater wetland taxa is limited to a ~1-2 m zone adjacent to the main waterbody (Figure 7). Because water level fluctuation due to tides is negligible over most of the system (under 0.5 m in the main tidal creek at Moon Pond and only a few cm or less in the rest of East Harbor), wave action is presumably the cause of mortality in this zone. On a broader scale, the lack of penetration of saline water into the far reaches of these marshes has allowed most of the existing vegetation to survive. In fact, root zone salinities measured along transects 1 and 2 are ≤ 5 ppt only meters away from the marsh edge (see Appendix II). Thus, it appears that a considerable volume of fresh groundwater is able to infiltrate the peripheral marshes from the surrounding uplands, thereby allowing freshwater species to persist. Until the tidal amplitude of the main water body is increased, the restoration of emergent salt marsh vegetation beyond the Moon Pond area may prove difficult.

Figure 7. Salt-killed zone of vegetation along the north shoreline of East Harbor (2004).

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Appendix I: Chronology of Events at Pilgrim Lake (East Harbor).

(J. Portnoy, Cape Cod National Seashore)

1845	Bridge built across East Harbor inlet to connect Truro and Provincetown.
1868	East Harbor inlet filled, purportedly to block sand from filling Provincetown Harbor.
1873	Railway built across dike to service Provincetown.
1894	Drainage system constructed at southeast end of Lake (Moon Pond) connecting Lake to Cape Cod Bay with flume at High Head Road.
1911	Carp found in lake, origin unknown , perhaps introduced as bait (Mass. Survey of Inland Waters).
1920s	Route 6A built on western side of railroad.
1935	WPA reconstructs existing (1894) pipe between Lake and Bay.
1952-3	Highway 6 built on fill along southern side of lake.
1956-8	Mass. Division of Waterways builds culvert and tide gate system to lower lake level for mosquito control with weir in place of 1894 flume at High Head Road.
9/1968	Fish kill (carp & white perch) following removal of weir boards, lowered lake level and likely oxygen depletion in remaining water. Salinity 10-18 0/00.
	Chironomid midge problem.
5/1969	9000 alewives (spawners) stocked in lake to consume midge larvae by Mass. Division of Fish and Game and Division of Marine Fisheries
6/1969	NPS applies Abate as midge larvicide by helicopter.
8/1969	NPS applies Abate as midge larvicide by helicopter.
5/1970	6000 more alewives stocked by DMF.
	NPS applies Abate as midge larvicide by helicopter.
1976	Mass. DPW repairs tide gates and pipe to bay.
5/1982	Adult alewives observed swimming over weir into lake.
9/2001	Fish kill includes > 30,000 juvenile alewives and hundreds of white perch, likely due to oxygen depletion.
12/6/01	NPS and Town of Truro experimentally open culvert valves.
1/25/02	NPS & Truro Health Board plan for bacteria monitoring at culvert & Beach Point.
2/27/02	Valves closed to allow salinity to decline for anadromous fish spawning per agreement with DMF.

5/02 NPS sets up vegetation and water quality monitoring plots in Moon Pond and East Harbor wetlands

6/24/02 Valves opened by Truro DPW and NPS.

7/4/02 Valves closed by Truro DPW and NPS due to Board of Health concerns for high *Enterococcus* at Beach Point. Pilgrim Lake salinity declines to 10 ppt.

7/11/02 Boards put in weir by Truro to dampen discharge and see if it helps with bacteria problem.

8/2002 Large midge (chironomid) hatch from Pilgrim Lake.

11/4/02 Valves opened and weir boards removed by Truro DPW and NPS per Conservation Commission and Selectmen decision.

6/2003 Salinity reaches 20-25 throughout Harbor;

Widgeon grass proliferates in Pilgrim Lake.

9/2003 Ten species of estuarine fish and crustaceans reestablish in East Harbor;

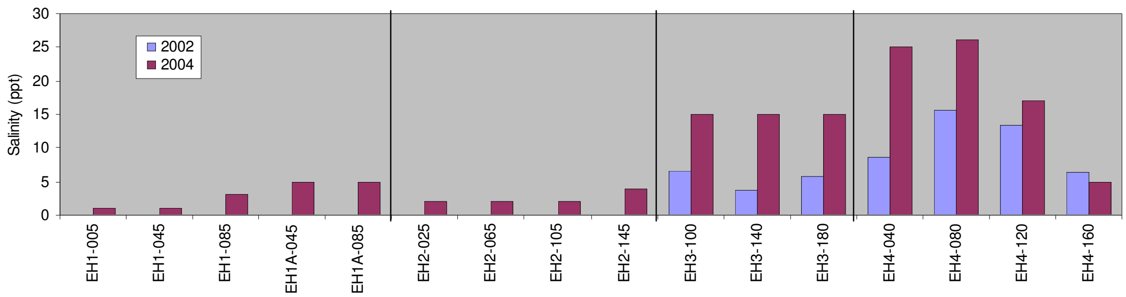
5/2004 Sand eels using East Harbor.

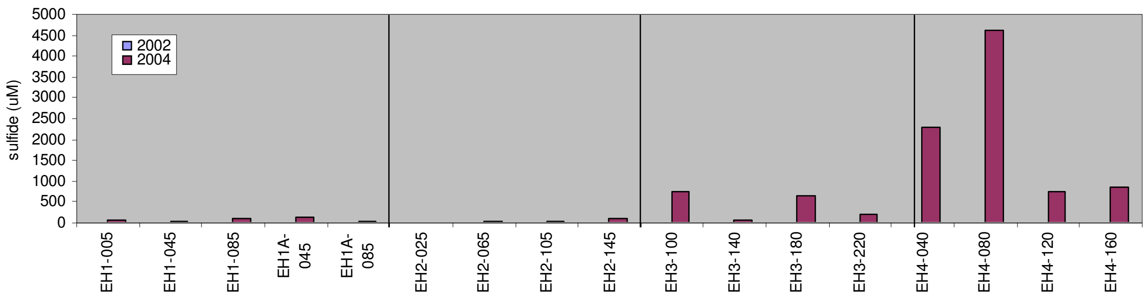
6/2004 Hard clams and steamers observed in sediments of former “Pilgrim Lake”;

Blue mussel bed develops in Moon Pond creek.

7/2004 Eelgrass discovered in Pilgrim Lake.

Appendix II. Change in root zone salinity and sulfide concentrations from June 2002 to June 2004 (porewater samples taken from a depth of 10 cm).





II. EAST HARBOR: SUBMERGED VEGETATION (SEAGRASSES)

Introduction

Following the 2002 opening of the culvert that links East Harbor with Cape Cod Bay, surface water salinities in open water have risen from ≤ 5 ppt to ~20 ppt in the winter and ~25 ppt in the summer. Plants and animals have responded rapidly to this change. One such response is the proliferation of seagrasses, particularly *Ruppia maritima* (widgeongrass). This species has grown rapidly since the onset of tidal restoration, forming thick beds across much of the littoral zone.

Seagrasses of all kinds are an important component of coastal ecosystems as they are vital to food webs and habitat structure. Some species, including *Ruppia*, are directly consumed by waterfowl. Seagrass detritus, along with its bacteria, provides food for a variety of worms, crabs, and certain filter feeders such as anemones and ascidians. The leaves themselves are excellent attachment sites for bryozoans, sponges, and hydroids as well as the eggs of ascidians and molluscs. In turn, these organisms are food for small fish which seek refuge in seagrass beds from larger predators. The importance of *Ruppia* to populations of benthic invertebrates in East Harbor has been previously demonstrated by Carlson (2003).

Increased *Ruppia* abundance in East Harbor may be related directly and/or indirectly to tidal restoration. This species is reportedly tolerant of salinities ranging from nearly fresh to hypersaline water (USGS 2004 and references therein) and it is difficult to know how much the elevated salinity level has contributed to enhanced productivity. It is noteworthy, however, that submerged vegetation (presumably *Ruppia*) is clearly evident in 1947 aerial photography when it was a freshwater system. Curiously, this was a time during which Asian carp – a species known to feed directly on widgeongrass (Sidorkewicj et al. 1998) – were known to be present. However, carp in East Harbor have occasionally been killed by severe oxygen depletions (Appendix II, previous section) and their populations have likely undergone major fluctuations. Notwithstanding, in subsequent photography (including images from 2000 and 2001) SAV cannot be detected, which corroborates on-the-ground observations by field personnel. Shortly after tidal exchange was facilitated in 2002, the carp were virtually eliminated from the system due to the salinity change. The absence of grazing pressure may now be allowing *Ruppia* to recover and subsequently flourish. Eelgrass (*Zostera marina*) has also appeared in places, although to a much lesser extent.

In the summer of 2003, it became obvious that submersed aquatic vegetation (SAV) was responding to tidal restoration at a rate and spatial scale that was likely to benefit estuarine organisms (see above). Thus, our objective in 2004 was to collect baseline data on SAV distribution and abundance within East Harbor so that temporal changes could be monitored and population dynamics better understood.

Methods

General Mapping and Inventory - To map seagrass beds, all shallow water areas (< 1m depth) were explored by canoe and kayak using low level (oblique angle) aerial photography as a reference. At each bed encountered, a GPS point was acquired and the following data recorded:

- Maximum length of bed (m)
- Maximum width of bed (m)
- Water depth at center of bed (m)
- Shoot density (high, medium, low)
- Species (*Ruppia maritima*, *Zostera marina*)
- Canopy at Surface? (Y/N)

An approximate area cover for each bed was estimated using the formula to calculate the area of an ellipse, which is: Pi*a*b, where a is ½ the length of the longest diameter axis (measured at its greatest width) and b is ½ the length of the shortest diameter axis (measured at its greatest height). Turbidity of the water column was also measured by Secchi disc at three separate locations (Figure 1).

Transects – Two parallel 50 m transects were established parallel to the shoreline at 5 different sites. The sites were chosen by randomly selecting 5 mapped *Ruppia* beds as start point locations. At each site, a nearshore (shallow) transect was established approximately 3-4 m away from the shoreline and run in a straight line roughly parallel to the shore. The exact placement of this transect was somewhat subjective in that it had to be oriented to avoid crossing land. A second (deep) transect was placed 5m offshore from the first in deeper water (Figure 2). PVC markers delineating the end points of each transect were pounded into the bottom sediment and GPSed for future reference. This site layout follows the design of Short et al. (2002).

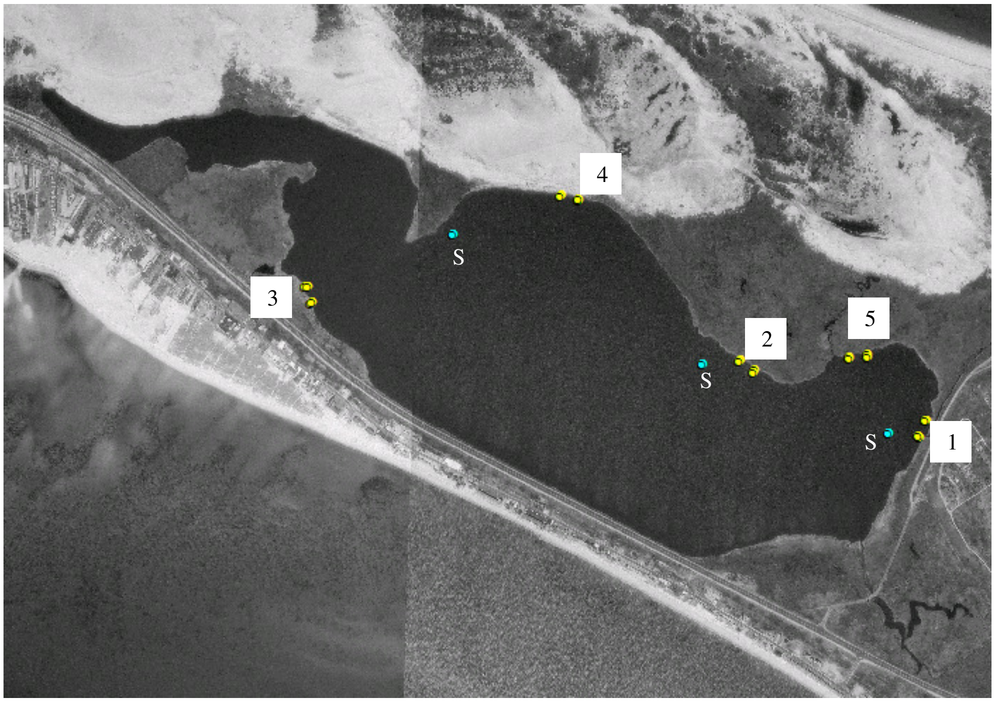


Figure 1. Transect endpoints for seagrass transects and Secchi disc sampling sites ("S").

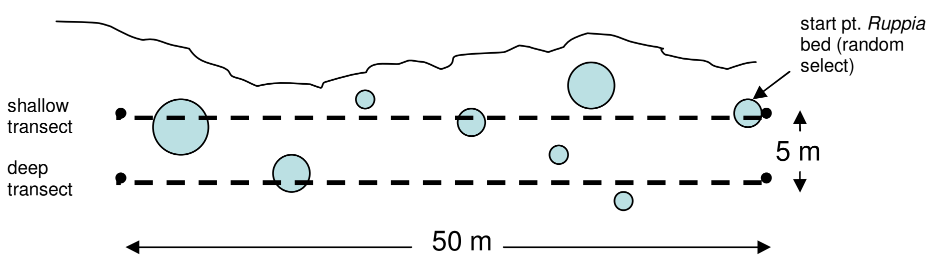


Figure 2. Schematic of transect layout for an individual site.

Seagrasses along these transects were characterized by point-intercept along the transect lines. At each 1m interval, from 0 to 50 m, the observer would peer through a view-scope to determine whether any live material came in contact with a vertical rod placed at each meter mark. The sum of all "hits" along the 50 m line was then used to calculate a percent cover value. In addition, the height of the tallest shoot on any plant touching the rod was measured with a meter stick.

Results

General Mapping and Inventory – A total of 203 discrete seagrass beds, almost all of which were *Ruppia*, were mapped. The beds were restricted to the perimeter of the system and were completely absent from the southern boundary adjacent to Route 6, where the shoreline is comprised of deteriorating asphalt and cobble originally placed there (1952) to prevent bank erosion. One cluster was found approximately 250 m from shore where a small shoal exists.

Unfortunately, it was not possible to accurately map the northwest cove as *Ruppia* formed more or less a continuous bed across this area. Exclusive of this cove, *Ruppia* beds ranged in size from 0.1 to 250m^2 (Figure 3). The minimum and maximum water depths recorded within the beds were 0.12 and 0.65m, respectively. *Zostera* was generally present in small numbers interspersed with widgeongrass along the shoreline of northeastern corner of the system (Figure 4). At 3 out of 14 sites where *Zostera* was found, there were small (> 15m^2), low density monospecific beds. Secchi depth readings from 3 open water sites in open water averaged 0.68m.

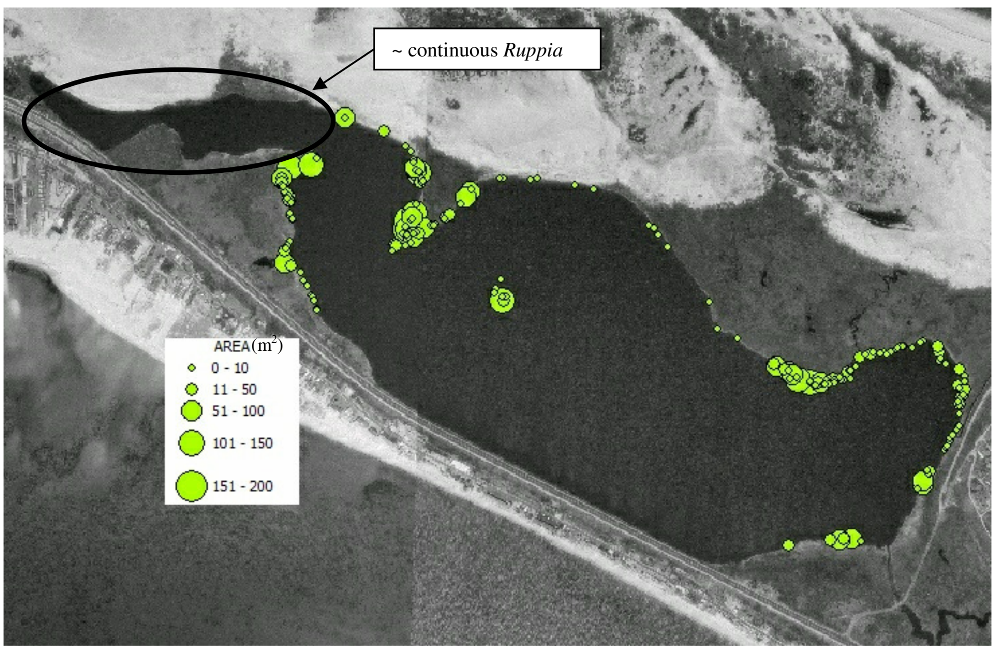


Figure 3. *Ruppia* beds in East Harbor (2004) (circle size is proportional to area cover as indicated in figure legend).

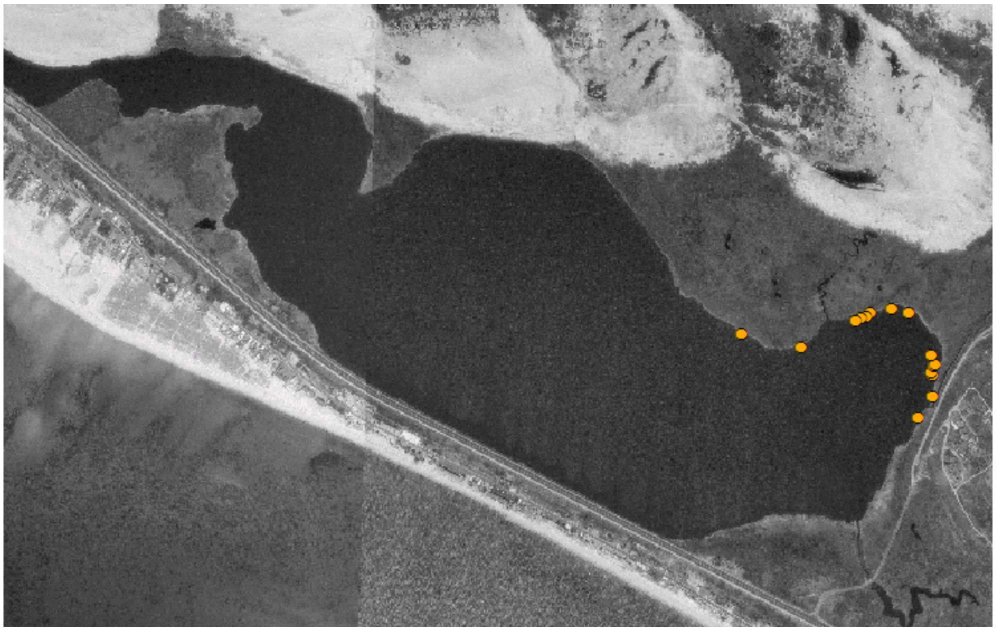


Figure 4. *Zostera* sites in East Harbor (2004).

Transects – Along the transects, *Ruppia* comprised the vast majority of seagrass cover, ranging between 6 and 34% among shallow transects and 2 and 20% among the deep transects. *Zostera* was present on only one transect with a total cover of 2%. Plants were significantly taller in shallow compared to deep transects, averaging 35 and 23 cm, respectively. In terms of water level, shallow transects averaged 40 cm while deep transects averaged 52 cm.

Table 1. Percent cover of seagrass, mean water depth, and mean shoot height by species and transect (2004)

	Transect	R. maritima	Z. marina	H_2O (cm)	Shoot ht (cm)
shallow	1	10%	2%	36	46
	2	22%	0%	41	56
	3	34%	0%	33	28
	4	6%	0%	44	22
	5	26%	0%	48	23
deep	1	4%	0%	45	37
	2	2%	0%	50	26
	3	20%	0%	55	18
	4	8%	0%	53	17
	5	6%	0%	59	18
	mean shallow	20%	0.4%	40	35
	mean deep	8%	0.0%	52	23

Discussion

Overall, the mapping effort provided some useful information on the distribution and abundance of seagrass populations within East Harbor. However, estimations of size were quite difficult for beds that were irregularly shaped. Furthermore, mapping may become more difficult over time as beds coalesce and there are fewer discrete patches. If this happens, it may be problematic to repeat this procedure. As an alternative, low-level, oblique-angle aerial photography, while not able to provide a quantitative measure of change, can be used to document gross, qualitative changes in seagrass cover across the entire system. Notwithstanding, the permanent transects will serve as a useful tool for quantitatively monitoring temporal variability. Because seagrass coverage along the line transects is currently very low, the transects should adequately capture any significant expansion, especially along deep transects where cover is currently near zero.

The elimination of Asian carp may have been the critical event that has allowed *Ruppia* to flourish in East Harbor. To some extent, *Zostera* may also be responding to increased salinity as this species prefers a range of 10 to 39 ppt (Davison and Hughes 1998). In the absence of grazing pressure, light limitation appears to be the primary factor regulating seagrass abundance at this time. Throughout the system, seagrasses were confined to very shallow waters < 70cm in depth (the limit of light penetration according to Secchi readings). In addition, mean shoot heights were significantly higher in the shallow compared to deep transects (Table 1).

Over time, the water clarity of East Harbor may improve. Considerable amounts of organic matter and fine sediments that accumulated during impoundment are now being exported during ebb tides. Perhaps more importantly, seagrasses themselves may increase water clarity by assimilating nutrients from both the sediment and directly from the water column, thereby reducing the nutrient supply for suspended phytoplankton. At the same time, root growth will stabilize the sediment, allowing less particulate matter to be re-suspended. This process is already evident in the northwest cove where seagrass biomass is maximal. Although water depths here exceed 150 cm in places, the bottom is clearly visible. The absence of carp has also improved sediment stability, as this fish is notorious for increasing turbidity as a benthic forager.

In theory, uptake of nutrients and sediment stabilization should result in a positive feedback loop whereby any reduction in light attenuation through seagrass growth will enhance conditions for further growth. It should be noted that with increased productivity, there is the potential for reductions in nighttime dissolved oxygen concentrations in the water column. However, it is unknown how much biomass might produce such a change in water quality. In addition, should the tidal regime be restored further, this effect would be mitigated by an increased volume of oxygen-rich water entering East Harbor on flood tides.

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III. HATCHES HARBOR

Introduction

Since 1997, when a series of adjustable culverts were installed through a tide-restricting dike, approximately 100 acres of Hatches Harbor has been undergoing hydrologic restoration. During this time, the formerly restricted portion of the marsh has exhibited dramatic changes in the physical, chemical, and biological landscape (Farris and Portnoy 2000, Farris et al. 2002, Portnoy et al. 2003). This report focuses on vegetation data collected in the summer of 2004 as part of a continuing effort to evaluate the restoration process.

Methods

All materials and methods for vegetation monitoring are described in Smith 2004 and follow the design of Roman et al. 2001 and Neckles et al. 2002. As in previous years, vegetation was monitored by point intercept in the 1m^2 permanent plots. In addition, stem heights and densities of live *Phragmites australis* were recorded in all plots where it occurred.

Data analysis - A comprehensive analysis of community-level vegetation changes from 1997-2004 is presently being conducted by Charles Roman (North Atlantic Coast Cooperative Ecosystem Studies Unit, University of Rhode Island). Accordingly, this report summarizes the responses of individual key species in the system. *Spartina* and *Suaeda* spp. cover data are plotted as histograms showing temporal change within individual plots. *Phragmites* cover data were subjected to linear regression to examine relationships with environmental variables. Standard errors corresponding to *Phragmites* stem height values are shown to indicate the statistical relevance of temporal changes. ARCGIS ver. 8.0 was used to depict spatial patterns of *Phragmites* biomass.

Results

Trends in the distribution and abundance of key species - *S. alterniflora* and *S. patens* cover have undergone dynamic change since 1997. A multiyear comparison of *S. alterniflora* shows that considerable variation in cover has occurred, with both losses and gains over time (Figure 1). The most plausible explanation for these patterns is that hydrologic conditions continue to change as flow through the culverts is increased with each incremental opening. Conditions of unrestricted “low marsh” habitat are now occurring across more of the restricted marsh. *S. alterniflora* appears to be following this change, becoming established initially near the creek bank and then expanding outward as evidenced by the many hundreds of new seedlings that have germinated beyond the boundaries of the existing population (Figure 2). However, some of the creek bank plants are now being lost due to changes in the morphology of the main tidal creek (e.g., plot

HH2A-000; Figure 1). More specifically, parts of the channel have widened considerably as a result of increased water velocities.

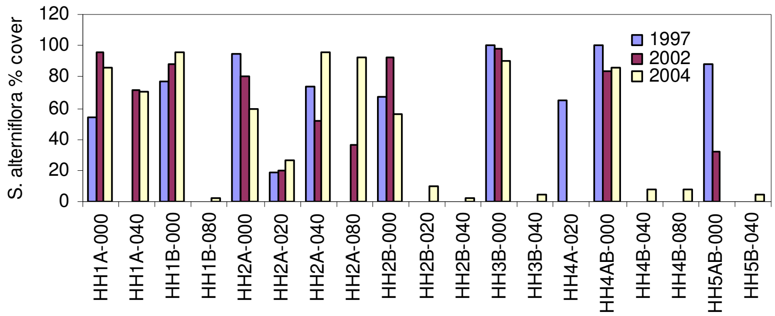

Figure 1. Comparison of *S. alterniflora* cover in 1997, 2002, and 2004 (restricted side only).

Figure 2. Seedlings of *S. alterniflora* germinating on a mudflat amidst stunted and dead *Phragmites* stems.

S. patens is also expanding into areas distant from the main tidal creek as saltwater begins to reach further into the marsh (Figure 3). Like *S. alterniflora*, it is also being lost in places along the downslope edge, primarily because its optimal range of salinity/hydroperiod conditions are moving further upslope. In general, the low marsh/high marsh interface is shifting away from the main tidal creek with each stepwise increase in flow, with concurrent gains and losses on the upslope and downslope edges of these communities.

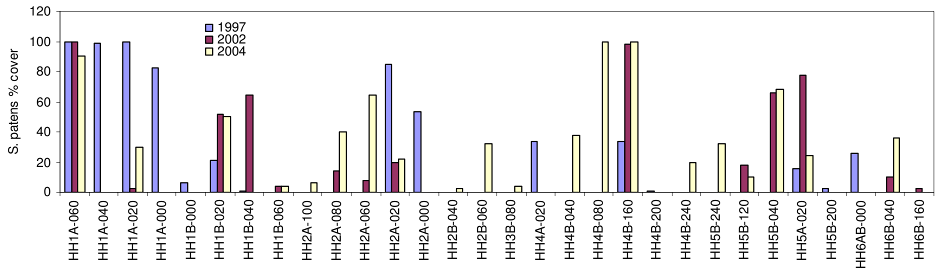

Figure 3. Comparison of *S. patens* cover in 1998 and 2004 (restricted side only).

Similar to *Spartina* spp., other native halophytes are expanding their range. In 2004, two species of sea blites (*Suaeda maritima* and *Suaeda linearis*) were observed along all 6 transects above the dike despite the fact that none were recorded prior to the first culvert openings in 1998 (Figure 4). *Suaeda* spp. have recently emerged in high numbers amongst the culms of beachgrass that were alive and well in 2003 but salt-killed in 2004 (Figure 5).

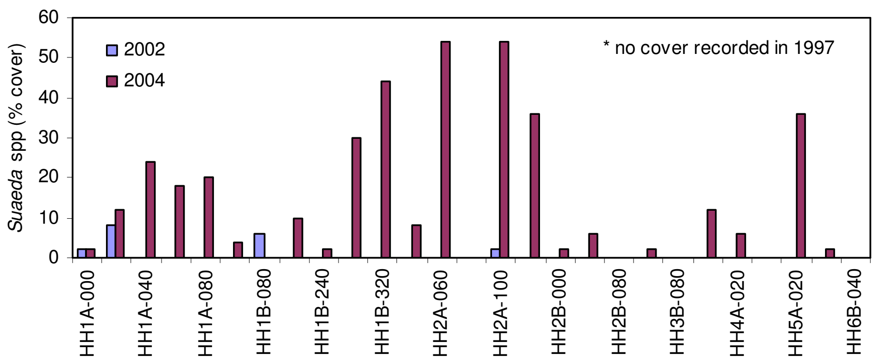

Figure 4. Percent cover of previously absent *Suaeda* species in 2002-2004 (restricted side only).

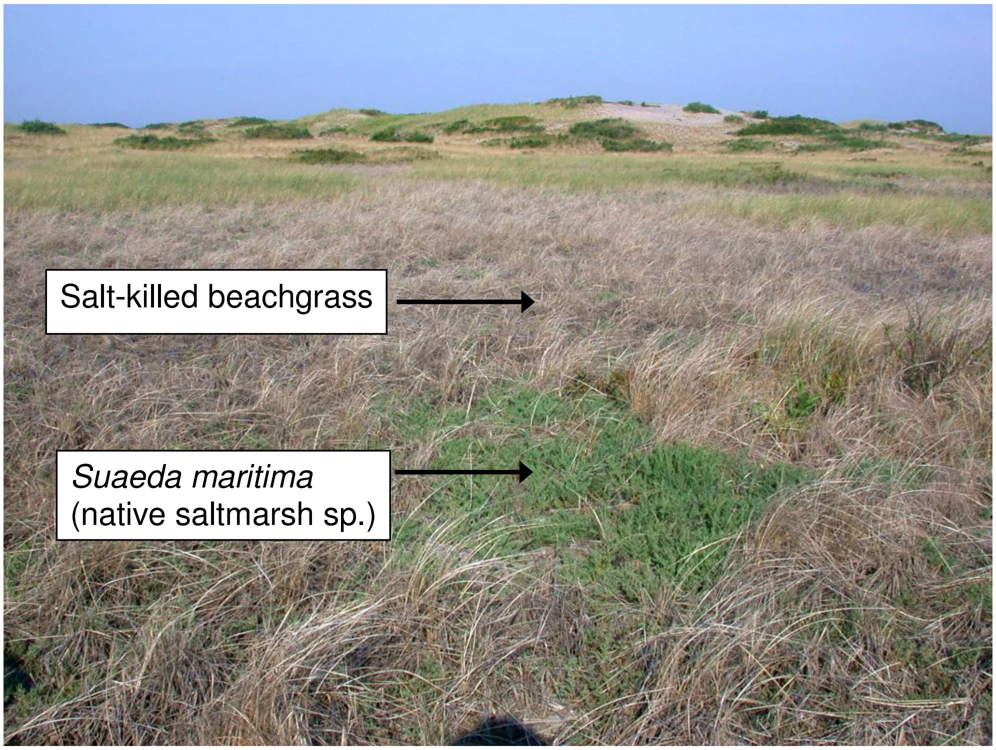

Figure 5. Expansion of wetland area since 2003 as indicated by the death of upland dune species and the proliferation of native salt marsh taxa (*Suaeda* spp.).

Changes in the area cover of *Phragmites* from 2002-2004 were significantly correlated with plot distance from the culverts ($F=12.2$, $p=0.0009$). The same was true for elevation ($F=15.8$, $p=0.0001$) but not 2004 porewater salinity (Figure 6). This indicates that instantaneous measures of root zone salinities do not always adequately explain patterns in *Phragmites* cover. Instead the influence of salinity over long periods of time may better reflect distribution and abundance – a metric that is intrinsically tied to distance from the culverts and elevation. The scatter of the data about the trend lines may be due to the confounding influence of hydroperiod since many closer, low-elevation plots may now be draining better, thereby mitigating some of the salinity effects. Conversely, *Phragmites* may be stressed by the drier conditions of the highest elevation plots in fresher parts of the system distant from tidal creeks. In addition, salinities continue to change with each enlargement of the culvert openings. As seawater penetrates farther and farther into the marsh, the cover responses (which may not be a good representation of plant biomass since there is no vertical component to this parameter) of salt tolerant vegetation may lag behind those of the physico-chemical environment.

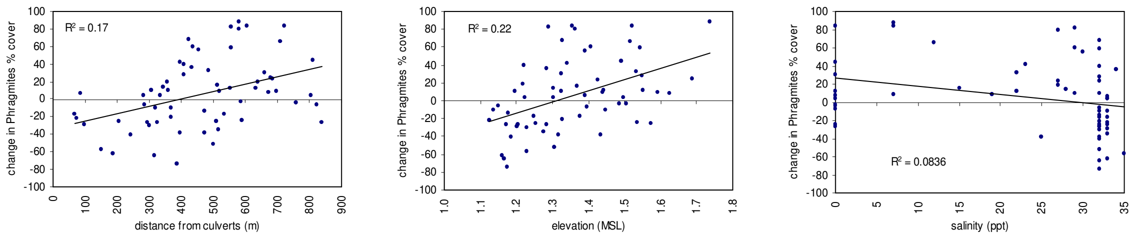

Figure 6. Change in *Phragmites* percent cover vs. distance, elevation, and 2004 porewater salinity.

Compared to area cover, suppression of height growth appears to be a much more rapid response to increased tidal flow. From 2002 to 2004, for example, *Phragmites* stem heights continued to decrease in the majority of plots along transect 2 (Figure 7). As in previous years, the magnitude of reduction was highest in plots closest to the main tidal creek. Conversely, *Phragmites* has recently appeared between plots HH2B-260 and -320, where it was previously absent.

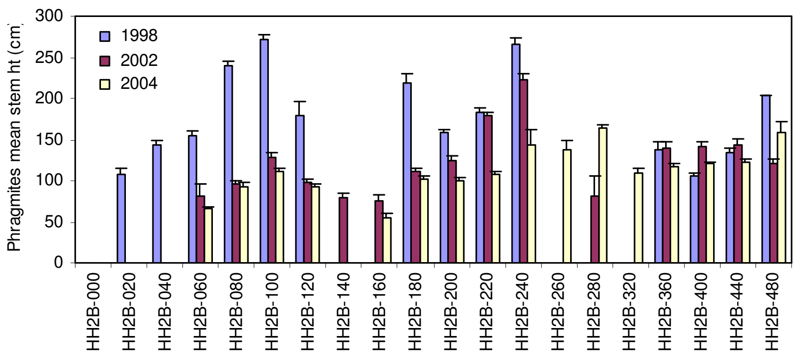

Figure 7. Change in *Phragmites* mean stem length along transects 1 and 2 from 2002-2004.

Phragmites biomass (estimated from stem densities and stem heights as per Thursby et al. 2002) in 2004 showed a clearly discernible pattern with respect to porewater salinity, even though this relationship was not statistically significant by regression analysis (Figure 8). Notwithstanding, biomass generally peaked where salinities were lower than full-strength seawater, but still high enough to exclude other salt-intolerant taxa. In this way, *Phragmites* seems to be exploiting a niche where salinity is tolerable, but high enough to eliminate interspecific competition. However, chronic exposure to elevated salinities and the gradual encroachment of native salt marsh vegetation into these areas are likely to eventually affect *Phragmites* in a negative way.

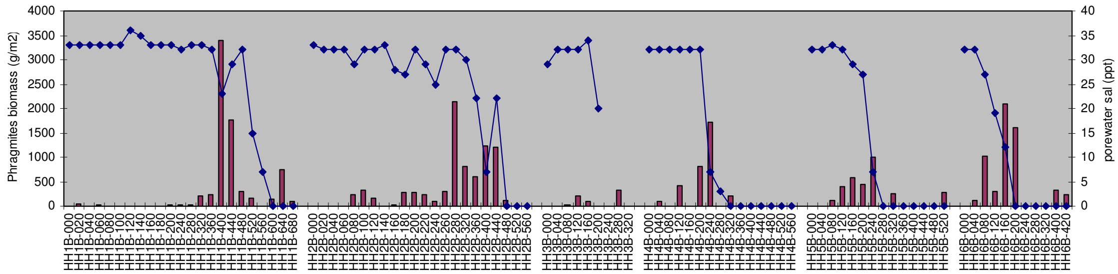

Figure 8. 2004 *Phragmites* biomass and porewater salinity by plot (numeric values in plot names indicate distance from main tidal creek).

From a marsh-wide perspective, *Phragmites* is responding to increased tidal influence primarily by shifts in its distribution (Figure 9). While *Phragmites* near the main tidal creek is disappearing or exhibiting stunted growth, the far edge of the population is encroaching upon the freshwater wetland community behind it. In other words, Zone 2 (see Figure 10) of the population is moving away from the creek bank and toward the freshwater-upland transition zone, although the degree to which this is occurring is different along each transect. The largest change has occurred along transects 1 and 2, which experience the greatest amount of seawater influence owing to their close proximity to the culverts.

Figure 9. Changes in *Phragmites* biomass (estimated from stem heights and densities) from 2002-2004 (arrow indicates direction of movement of peak biomass).

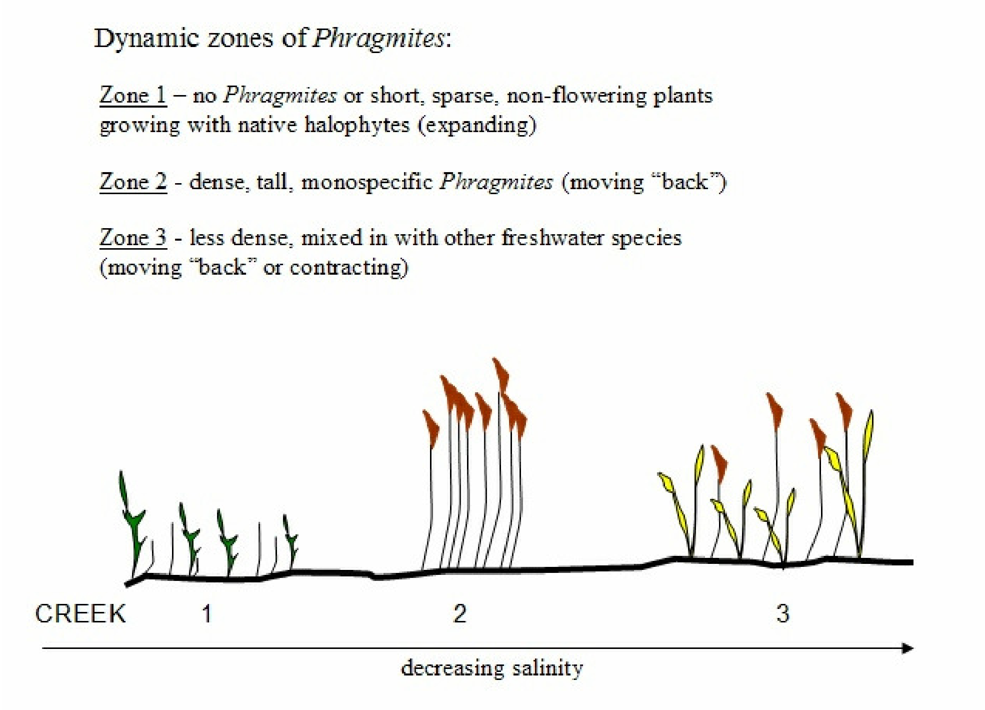

Figure 10. Schematic diagram of zones defining *Phragmites* population vigor and abundance.

Discussion

The vegetation community in Hatches Harbor continues to exhibit significant responses to incremental restoration of tidal flow. The most recent enlargement of culvert cross sectional area in October 2003 resulted in a notable increase in land surface area subject to inundation. This resulted in the death of salt-intolerant wetland and upland plants, coinciding with the establishment of native halophytes in these areas.

The *Phragmites* population continues to "migrate" away from the main tidal creeks toward the freshwater wetland-upland boundary. In effect, this has allowed salt marsh vegetation, including *Spartina* and *Suaeda* spp., to greatly expand their range. The plethora of *S. alterniflora* seedlings that emerged in areas originally occupied by dense *Phragmites* speaks to the magnitude of change.

Overall, a substantial portion of the formerly-restricted marsh now resembles the unrestricted landscape with respect to vegetation communities. Moreover, as restoration proceeds it is expected that there may be some positive feedback whereby the establishment of desirable taxa in new places enhances their capacity for further expansion by the production of new seeds and propagules. In turn, this may eventually result in enough competitive pressure to encourage further withdrawal of the *Phragmites* population. Any further increase in tidal flow would presumably enhance the rate and magnitude of these kinds of changes.

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IV. HERRING RIVER

Introduction

The Herring River is the largest hydrologically-impaired estuarine/riverine system within Cape Cod National Seashore. Restriction of tidal exchange occurred after the installation of a dike across the mouth of the river in 1909 and the area has since suffered from a myriad of problems typical of other diked marshes (Portnoy 2003). Currently, the Seashore is working with the Town of Wellfleet, Massachusetts Coastal Zone Management, and several federal agencies to restore tidal flow to the system. In anticipation of this effort, pre-restoration data on vegetation communities were acquired in the summer/fall of 2004 to provide a baseline for long-term monitoring. This report presents and discusses this information.

Methods

The materials and methods used for Herring River vegetation monitoring follow those described in previous sections of this document (Hatches Harbor and East Harbor monitoring), all of which are based upon Roman et al. (2001) and Neckles et al (2002).

A total of 14 transects (73 plots) running from marsh edge to upland were established in unrestricted (n=4) and restricted (n=10) marsh areas on either side of the Herring River dike. No transects were established above High Toss Road (see Figure 1) in 2004 although there are plans to do so in 2005. Nine of these transects were positioned in an attempt to repeat vegetation surveys done in 1975 (Snow 1975) and 1978 (Gaskell 1978), while the other five were randomly located. On each transect, the first plot was randomly located within the first 2 m of the marsh edge. From there, a second and third plot were located 10 and 20m upslope (toward the upland), perpendicular to the marsh edge. Plots were then put in every 20 m until the upland was reached.

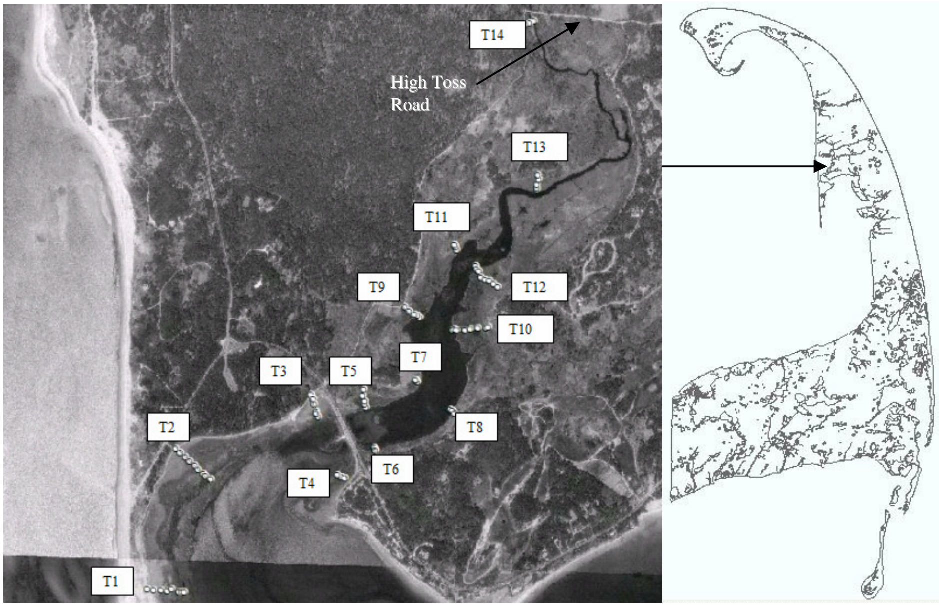

Figure 1. Map of Herring River transects and plots (2004).

Plant cover by species was recorded by point intercept in each plot. Where *Phragmites* was present, the heights and numbers of all live stems were recorded.

Data analysis – Although previous data exists on the widths of various vegetation bands along transects (e.g., distance from marsh edge to *Phragmites* boundary occupied by *S. alterniflora*), it was not possible to relocate the original transect positions with a degree of accuracy necessary to compare data (GPS technology was not available at the time of earlier studies). The current vegetation community exhibits spatial heterogeneity in zonation on a scale much finer than the potential error between original vs. relocated sites. Thus, the 2004 data will serve as the baseline dataset for the long-term monitoring of this restoration project and there is limited data analysis. Principle Components Analysis (PCA) of log-transformed percent cover values was used to portray taxonomic heterogeneity across the monitoring network. Trends in *Phragmites* canopy height were analyzed by linear regression. ARCGIS ver. 8.0 was used to depict spatial patterns of *Phragmites* biomass.

Results

From the standpoint of community-level variability, the vegetation above the dike shows a spatial gradient of dissimilarity compared with that below the dike (Figure 1). In general, divergence from the unrestricted community grows larger with distance above the dike and distance from the marsh edge.

A total of 55 species were found among all plots. Plots above the dike (i.e., the restricted portion of the system) were significantly more diverse than those below the dike (reference marsh), mainly due to the presence of various freshwater species (Figure 2). *Spartina alterniflora* occupies the marsh edge immediately upstream of the dike. This zone narrows upstream and finally disappears near transect 11. Beginning in the area upstream of transect 9, *Phragmites* dominates the wetland vegetation, reaching heights of up to 3.5 meters. Nearer to High Toss Road, *Phragmites* declines and gives way to *Typha angustifolia* (narrow-leaved cattail) and other freshwater taxa. This change coincides with a porewater salinity gradient although high sulfide concentrations along portions of transects 9 and 10 excludes *Phragmites* from what otherwise appear to be suitable salinity conditions (Table 1).

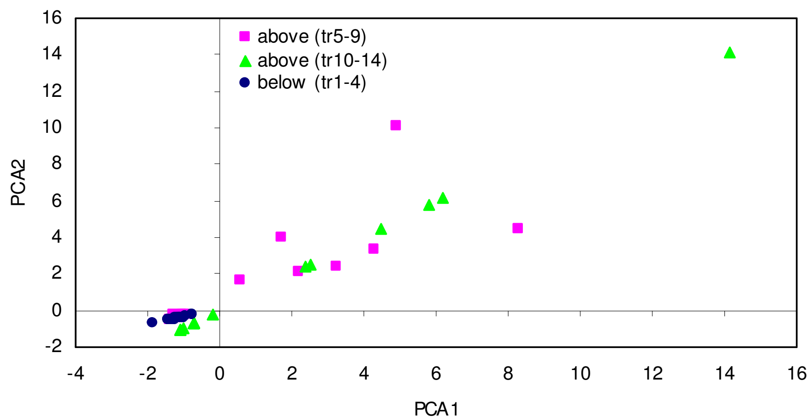

Figure 1. PCA of community composition showing dissimilarity between plots (distinguished by groupings of transects) below (unrestricted) and above (restricted) the Herring River dike.

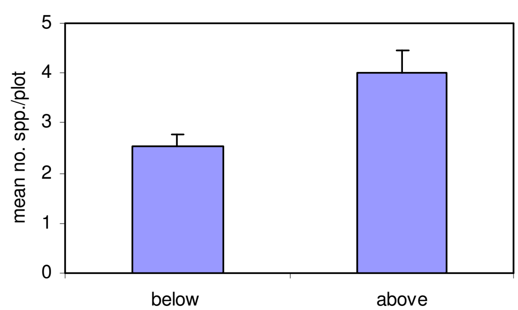

Figure 2. Species richness of plots below (unrestricted) and above (restricted) the Herring River dike.

Table 1. Pore-water quality by transect (all plots pooled) (J. Portnoy, unpublished data).

Transect	Sal (ppt)	pH	Alk (mEq)	Sulfide (uM)	
1	34	5.0	3.8	396	
2	32	6.1	3.2	218	
3	25	5.3	2.5	9	
4	29	4.8	1.8	48	unrestricted
5	32	5.3	4.0	851	restricted
6	29	2.1	0.7	15	
7	29	2.1	1.1	37	
8	26	2.2	1.4	181	
9	23	5.3	4.1	860	
10	23	6.6	5.8	1131	
11	20	4.3	2.2	302	
12	21	5.6	2.2	52	
13	8	4.9	1.0	1	
14	0	5.9	0.9	30	

Mean stem heights of *Phragmites* ranged between 104 and 312 cm, with values generally increasing with distance above the dike (regression line in Figure 3). Variance from this trend is mainly due to the shorter stature of *Phragmites* near the river or the upland edges.

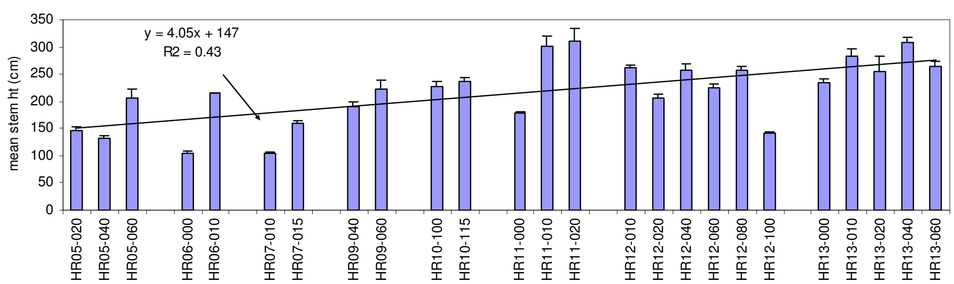

Figure 3. Mean stem heights of *Phragmites* by plot (trendline represents linear regression of height vs. plot; numeric values in plot names represent distance away from river edge).

Phragmites biomass, estimated from stem heights and densities (Thursby et al. 2002), exhibited a similar trend with the highest values occurring along transects 11, 12, and 13. No *Phragmites* occurs below the dike.

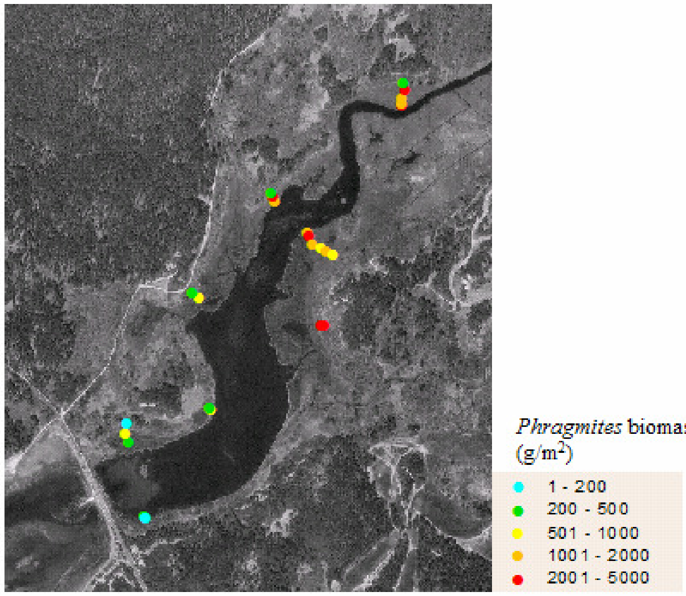

Figure 6. Estimated biomass of *Phragmites* along transects above the dike.

Discussion

The vegetation of the restricted portion of Herring River exhibits a spatial trend in species composition that largely reflects the current salinity gradient. Immediately above the dike, a vigorous zone of *S. alterniflora* still remains, although in a few places where there is inadequate drainage and high sulfide concentrations (e.g., transect 10) the leaves are extremely chlorotic. Proceeding upstream, the *S. alterniflora* zone rapidly narrows and disappears approximately 750 m away from the dike, where it is replaced by dense, monospecific *Phragmites*. Similar to East Harbor and Hatches Harbor, *Phragmites* in this zone appears to be exploiting a niche where salinities are high enough to exclude all other freshwater taxa, but low enough to produce little physiological stress. At points further upstream, where conditions are fresher, *Phragmites* gives way to *Typha* and other various freshwater species. Tidal restoration would presumably result in expansion of S. *alterniflora* and other native halophytes, coinciding with movement of *Phragmites* and *Typha* zones in an upstream direction.

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